Plant and Soil **237:** 287–307, 2001.

Phosphorus loss from land to water: integrating agricultural and environmental management

Andrew N. Sharpley*, Richard W. McDowell & Peter J. A. Kleinman
*USDA-ARS, Pasture Systems and Watershed Management Research Unit Building 3702, Curtin Road, University Park, PA 16802-3702, USA *Corresponding author: Tel: +1-814-863-0948; Fax: +1-814-865-2058; E-mail: ans3@psu.edu*

Key words: eutrophication, fertilizers, manures, nonpoint source pollution, runoff, water quality

Abstract

Phosphorus (P), an essential nutrient for crop and animal production, can accelerate freshwater eutrophication, now one of the most ubiquitous forms of water quality impairment in the developed world. Repeated outbreaks of harmful algal blooms (e.g., *Cyanobacteria* and *Pfiesteria*) have increased society's awareness of eutrophication, and the need for solutions. Agriculture is regarded as an important source of P in the environment. Specifically, the concentration of specialized farming systems has led to a transfer of P from areas of grain production to animal production. This has created regional surpluses in P inputs (mineral fertilizer and feed) over outputs (crop and animal produce), built up soil P in excess of crop needs, and increased the loss of P from land to water. Recent research has shown that this loss of P in both surface runoff and subsurface flow originates primarily from small areas within watersheds during a few storms. These areas occur where high soil P, or P application in mineral fertilizer or manure, coincide with high runoff or erosion potential. We argue that the overall goal of efforts to reduce P loss to water should involve balancing P inputs and outputs at farm and watershed levels by optimizing animal feed rations and land application of P as mineral fertilizer and manure. Also, conservation practices should be targeted to relatively small but critical watershed areas for P export.

Introduction

Phosphorus (P), an essential nutrient for crop and animal production, can accelerate freshwater eutrophication (Carpenter et al., 1998; Sharpley, 2000). Recently, the US Environmental Protection Agency (1996) and US Geological Survey (1999) identified eutrophication as the most ubiquitous water quality impairment in the US. Eutrophication restricts water use for fisheries, recreation, and industry due to the increased growth of undesirable algae and aquatic weeds and oxygen shortages caused by their death and decomposition. Also, an increasing number of surface waters have experienced periodic and harmful algal blooms (e.g., *Cyanobacteria* and *Pfiesteria*), which contribute to summer fish kills, unpalatability of drinking water, formation of carcinogens during water chlorination and links to neurological impairment in humans (Burkholder and Glasgow, 1997; Kotak et al., 1993).

Although concern over eutrophication is not new, there has been a profound shift in our understanding of, and focus on, sources of P in water bodies. Since the late 1960s, the relative contributions of P to water bodies from point and non-point sources has changed dramatically. On one hand, great strides have been made in the control of point source discharges of P, such as the reduction of P in sewage treatment plant effluent. These improvements have been due, in part, to the ease in identifying point sources. On the other hand, less attention has been directed to controlling non-point sources of P, due mainly to the difficulty in their identification and control (Sharpley and Rekolainen, 1997). Thus, control of non-point sources of P is a major hurdle to protecting fresh surface waters from eutrophication (Sharpley and Tunney, 2000; Sharpley et al., 1999a; Withers et al., 2000).

While a variety of non-point sources, ranging from suburban lawns to construction sites to golf courses, contribute P to water bodies, agriculture, particularly intensive livestock agriculture, is receiving more and more attention (Lander et al., 1998; Sharpley, 2000).

This may be attributed to the evolution of agricultural systems from net sinks of P (i.e., deficits of P limit crop production) to net sources of P (i.e., P inputs in feed and mineral fertilizer can exceed outputs in farm produce). Before World War II, for example, farming communities tended to be self-sufficient in that they produced enough feed locally to meet animal requirements and could recycle the manure nutrients effectively to meet crop needs. As a result, nutrients were generally recycled in relatively localized areas. After World War II, farming systems became more specialized in the USA, with crop and livestock operations in different regions of the country. Today, less than a third of the grain is produced on farms where it is grown (Lanyon, 2000). This has resulted in a major one-way transfer of P from grain-producing areas to animal-producing areas (Sharpley et al., 1998b; Sims, 1997).

As animals inefficiently utilize P in feed (only 30% is retained), most of the P entering livestock operations ends up in manure, which is usually land applied locally. Animal manure can be a valuable resource for improving soil structure and increasing vegetative cover, thereby reducing surface runoff and erosion potential. However, in many areas of concentrated animal production, manures are normally applied at rates designed to meet crop nitrogen (N) requirements and to avoid groundwater quality problems created by leaching of excess N. This often results in a build up of soil test P above amounts sufficient for optimal crop yields, which can increase the potential for P loss in runoff as well as in leachate (Haygarth et al., 1998; Heckrath et al., 1995; Sharpley et al., 1996).

The ultimate goal of agricultural and environmental P management is to balance P inputs to the farm with outputs in primary produce such that no excess P is applied and soil P concentrations are kept at an optimum level for agronomic performance and minimal environmental impact. However, because of the potential for major changes in agricultural management and negative economic impacts, it is necessary to explore short-term or temporary fixes. In the USA, this has led the Environmental Protection Agency (EPA) and the Department of Agriculture (USDA) to devise a joint strategy for sustainable nutrient management for animal feedings operations (AFOs; USDA-USEPA, 1999). This strategy proposes a variety of voluntary and regulatory approaches, whereby all AFOs develop and implement comprehensive nutrient management plans by the year 2008. An important part of this strategy outlines how acceptable application rates of P as mineral fertilizer or manure will be determined.

In the USA, agencies charged with developing these strategies (i.e., EPA and USDA) have challenged the scientific community to provide technical leadership in developing sound criteria that identifies the risk of P loss from agricultural land to water (Sharpley et al., 1999b). The aim of this paper is to present research on P loss from land to water and show how this information is being used to define and support P management strategies that maintain agricultural production and protect water quality. We will discuss those factors controlling P loss in the context of developing practical tools for agricultural and environmental P management.

Assessing the Risk for Phosphorus Loss

Water quality concerns have forced many states in the USA to consider developing recommendations for land application of P and watershed management based on the potential for P loss in agricultural runoff (Sharpley et al., 1996; USDA-USEPA, 1999). Currently, these recommendations center on the identification of a threshold soil test P level above which the enrichment of P in surface runoff is considered unacceptable (Table 1). Existing agronomic guidelines may not be appropriate for water quality protection, and agronomic soil testing data may need to be reinterpreted to address environmental objectives (Sims and Sharpley, 1998). Specifically, agronomic soil test interpretations (i.e., low, medium, optimum, high) are based on the expected response of a crop to P, and cannot be directly translated to estimates of environmental risk, such as runoff P enrichment potential.

Even when soil testing data are properly reinterpreted for runoff enrichment potential, they provide an incomplete assessment of the potential for P loss from a site, as such data do not account for processes controlling the transport of P in surface runoff and subsurface flow (Kleinman et al., 2000). For example, adjacent fields having similar soil test P levels, but differing susceptibilities to surface runoff and erosion due to contrasting topography and management, may have substantially different P loss potentials (Sharpley and Tunney, 2000).

Generally, most P exported from agricultural watersheds comes from only a small part of the landscape during a few relatively large storms, where hydrologically active areas of a watershed contributing surface

Table 1. Threshold soil test P values and P management recommendations (adapted from Lory and Scharf, 2000; Sharpley et al., 1996)

State	Environmental soil P threshold mg kg^{-1}	Soil test P method	Management recommendations for water quality protection
Arkansas	150	Mehlich-3	*At or > 150 Mg P kg^{-1}:* apply no P, provide buffers next to streams, overseed pastures with legumes to aid P removal, and provide constant soil cover to minimize erosion.
Colorado	100	Olsen	*> 100 Mg P kg^{-1}:* hog producers with >36,000 lbs capacity, no P applied unless runoff is minimal.
Delaware	50	Mehlich-1	*> 50 Mg P kg^{-1}:* apply no more P until soil is significantly decreased.
Idaho	50 & 100	Olsen	*Sandy soils > 50 Mg P kg^{-1}:* *Silt loam soils > 100 Mg P kg^{-1}:* apply no more P until soil P is significantly decreased.
Kansas	100–200	Bray-1	Regions of the state coincide with high (eastern) to low (western) runoff. Swine producers must eliminate manure applications above the threshold.
Ohio	150	Bray-1	*> 150 Mg P kg^{-1}:* decrease erosion and/or eliminate P additions.
Oklahoma	130	Mehlich-3	*30 B 130 Mg P kg^{-1}:* half P rate on slopes > 8%. *130 B 200 Mg P kg^{-1}:* half P rate and adopt measures to decrease surface runoff and erosion. *> 200 Mg P kg^{-1}:* P rate not to exceed crop removal.
Maine	40–100	Morgan	Apply no P in sensitive (40 Mg P kg^{-1}) and non-sensitive watershed (100 Mg P kg^{-1}).
Maryland	75	Mehlich-1	*Use P index >75 Mg P kg^{-1}:* soils with high index must reduce or eliminate P additions.
Michigan	75	Bray-1	*75 B 150 Mg P kg^{-1}:* P application should equal crop removal. *> 150 Mg P kg^{-1}:* apply no P from any source.
Mississippi	70	Lancaster	*> 70 Mg P kg^{-1}:* no P added
Texas	200	Texas A&M	*> 200 Mg P kg^{-1}:* P addition not to exceed crop removal
Wisconsin	75	Bray-1	*< 75 Mg P kg^{-1}:* rotate to P demanding crops and decrease P additions. *> 75 Mg P kg^{-1}:* discontinue P additions.

runoff to streamflow are coincident with areas of high soil P (Gburek and Sharpley, 1998; Pionke et al., 1997). To be most effective, risk assessment must consider 'critical source-areas'; areas within a watershed that are most vulnerable to P loss in surface runoff (Gburek and Sharpley, 1998). Critical source areas are dependent on the coincidence of transport (surface runoff, erosion, and subsurface flow) and site management factors (functions of soil, crop, and management) (Table 2). Transport factors mobilize P sources, creating pathways of P loss from a field or watershed. Site management factors are typically well defined and reflect land use patterns related to soil P status, mineral fertilizer and manure P inputs, and tillage (Table 2).

Even in regions where subsurface flow pathways dominate P transport, areas contributing P to drainage waters appear to be localized to soils with high soil P saturation and hydrological connectivity to the drainage network (Schoumans and Breeuwsma, 1997). Therefore, soil P levels alone have little meaning vis a vis P loss potential unless they are used in conjunction with estimates of potential surface runoff and subsurface flow.

Table 2. Factors influencing P loss from agricultural watersheds and its impact on surface water quality

Factors	Description
Transport	
Erosion	Total P loss strongly related to erosion.
Surface runoff	Water has to move off or through a soil for P to move.
Subsurface flow	In sandy, organic, or P-saturated soils, P can leach through the soil.
Soil texture	Influences relative amounts of surface and subsurface flow occurring.
Irrigation runoff	Improper irrigation management can induce surface runoff and erosion of P.
Connectivity to stream	The closer the field to the stream, the greater the chance of P reaching it
Channel effects	Eroded material and associated P can be deposited or resuspended with a change in stream flow. Dissolved P can be sorbed or desorbed by stream channel sediments and bank material.
Proximity of P-sensitive water	Some watersheds are closer to P-sensitive waters than others (i.e., point of impact).
Sensitivity P input	Shallow lakes with large surface area tend to be more vulnerable to eutrophication.
Site management	
Soil P	As soil P increases, P loss in surface runoff and subsurface flow increases.
Applied P	The more P (mineral fertilizer or manure), the greater the risk of P loss.
Application method	P loss increases in the order: subsurface injection; plowed under; and surface broadcast with no incorporation.
Application timing	The sooner it rains after P is applied, the greater the risk for P loss

Development of the phosphorus index

To overcome the limitations of using a soil P threshold as the sole measure of site P loss potential, the US Natural Resource Conservation Service (NRCS), in cooperation with research scientists, developed a site assessment tool for P loss potential (i.e., the P index, Table 3). The P index was designed as a screening tool for use by field staff, watershed planners, and farmers to rank the vulnerability of sites to P loss in surface runoff (Lemunyon and Gilbert, 1993).

Calculating site vulnerability to phosphorus loss

The P index accounts for and ranks transport and site management factors controlling P loss in surface runoff and sites where the risk of P movement is expected to be higher than that of others (Tables 3 and 4). Site vulnerability to P loss in surface runoff is assessed by selecting rating values for a variety of transport (Table 3) and site management factors (Table 4).

To calculate transport potential for each site, erosion, surface runoff, leaching potential, and connectivity values were first summed (Table 3). Dividing this summed value by 23, the value corresponding to 'high' transport potential (erosion is 7, surface runoff is 8, leaching potential is 0, and connectivity is 8), a relative transport potential was determined. This normalization process assumes that when a site's full transport potential is realized, 100% transport potential is realized. Thus, transport factors <1 represent a fraction of the maximum potential (Table 3).

Calculation of site management factors of the P index are based on the Mehlich-3 P concentration of surface soil samples collected at each site and P application as mineral fertilizer or manure as determined from annual farmer surveys (Table 4). The correction factor of 0.2 for soil test P is based on field data which

Table 3. Phosphorus loss potential due to transport characteristics in the P index

Characteristics	Relative Ranking					Field Value
Soil Erosion	Soil loss (Tonnes/ha/year)					
Soil Runoff Class	Very Low	Low	Medium	High	Very High	
	0	**1**	**2**	**4**	**8**	
Subsurface Drainage	Very Low	Low	Medium	High	Very High	
	0	**1**	**2**	**4**	**8**	
Leaching Potential	Low		Medium		High	
	0		**2**		**4**	
Connectivity	Not connected†		Partially connected‡		Connected§	
	0	**1**	**2**	**4**	**8**	

Total Site Value (sum of erosion, surface runoff, leaching, and connectivity values):
Transport Potential for the Site (total value / 23) ¶:

†Field is far away from water body. Surface runoff from field does not enter water body.
‡Field is near but not next to water body. Surface runoff sometimes enters water body, e.g., during large intense storms.
§Field is next to a body of water. Surface runoff from field always enters water body.
¶The total site value is divided by a high value (23).

showed a 5-fold greater concentration of dissolved P in surface runoff with an increase in mineral fertilizer or manure addition compared to an equivalent increase in Mehlich-3 P (Sharpley and Tunney, 2000).

A P index value, representing cumulative site vulnerability to P loss, is obtained by multiplying summed transport and site management factors (Table 5). The P index values are normalized so that the break between high and very high categories is 100. This is done by calculating a site P index value, assuming all transport and source factors are high. Erosion is set at 7 tonnes ha^{-1} considered a high value for Pennsylvania and soil test P is set at 200 mg kg^{-1} Mehlich-3 P, which is proposed as a non-site specific threshold for Pennsylvania (Beegle, 2000). The break between medium and high and low and medium is calculated using the same method and soil test P concentrations of 50 and 30 mg Mehlich-3 P kg^{-1}, respectively. These Mehlich-3 P levels correspond to crop response and fertilizer recommendations for Pennsylvania, with 50 mg kg^{-1} sufficient for production and no response to added P and 30 mg kg^{-1} the low value (Beegle, 2000).

Management interpretations of the phosphorus index

Since its inception, two major changes have been introduced to the P index. First, source and transport factors were related in a multiplicative rather than additive fashion, in order to better represent actual site vulnerability to P loss. For example, if surface runoff does not occur at a particular site, its vulnerability should be low regardless of the soil P content. In the original P index, a site could be ranked as very highly vulnerable based on site management factors alone, even though no surface runoff or erosion occurred. On the other hand, a site with a high potential for surface runoff, erosion or subsurface flow but with low soil P is not at risk for P loss, unless P as mineral fertilizer or manure is applied. Second, an additional transport factor reflecting distance from the stream was incorporated into the P index. The contributing distance categories in the revised P index are based on hydrological analysis. This analysis considers the probability (or risk) of occurrence of a rainfall event of a given magnitude which will result in surface runoff to the stream (Gburek et al., 2000).

In addition to its function as a practical screening tool, the P index can also be used to identify agricultural areas or management practices that have the greatest potential to accelerate eutrophication. As such, the P index will identify alternative management options available to land users, providing flexibility in developing remedial strategies. Some general recommendations are given in Table 6. In considering these recommendations, one should keep in mind that P management is very site-specific and requires a well-planned, coordinated effort between farmers, extension agronomists, and soil conservation specialists.

In its current form, the P index is not a quantitative predictor of P loss in surface runoff or subsurface flow from a watershed. Rather it is a qualitative assessment tool to rank site vulnerability to P loss. Ultimately, the

Table 4. Phosphorus loss potential due to site management characteristics in the P index

Site Characteristics	Relative Ranking					Field Value
	Very Low	Low	Medium	High	Very High	
Soil Test P			Soil test P (mg/kg)			
Loss Rating Value			Soil Test P ∗ 0.2			
Fertilizer P Rate			Fertilizer Rate (kg P/ha)			
Fertilizer Application Method and Timing	Placed with planter or injected more than 2" deep	Incorporated <1 week after application	Incorporated >1 week or not incorporated >1 following application in spring–summer	Incorporated >1 week or not incorporated following application in autumn–winter	Surface applied on frozen or snow covered soil	
	0.2	**0.4**	**0.6**	**0.8**	**1.0**	
Loss Rating Value	Fertilizer P Application Rate ∗ Loss Rating for Fertilizer P Application Method and Timing					
Manure P Rate			Manure application (kg P/ha)			
Manure Application Method and Timing	Placed with planter or injected more than 2" deep	Incorporated <1 week after application	Incorporated >1 week or not incorporated >1 following application in spring–summer	Incorporated >1 week or not incorporated following application in autumn–winter	Surface applied on frozen or snow covered soil	
	0.2	**0.4**	**0.6**	**0.8**	**1.0**	
Loss Rating Value	Manure P Application Rate ∗ Loss Rating for Manure P Application Method and Timing					
			Total Site Management Value (sum of soil, fertilizer, and manure P loss rating values):			

Table 5. Worksheet and generalized interpretation of the P index

P Index	Generalized interpretation of the P index
Low < 30	LOW potential for P loss. If current farming practices are maintained, there is a low probability of adverse impacts on surface waters.
Medium 30–70	MEDIUM potential for P loss. The chance for adverse impacts on surface waters exists, and some remediation should be taken to minimize the probability of P loss.
High 70–100	HIGH potential for P loss and adverse impacts on surface waters. Soil and water conservation measures and a P management plan are needed to minimize the probability of P loss.
Very high > 100	VERY HIGH potential for P loss and adverse impacts on surface waters. All necessary soil and water conservation measures and a P management plan must be implemented to minimize the P loss.

P index rating for a site = Transport potential value × Site management value/45 †. 145 is the value to normalize the break between high and very high to 100. The following is used:

Transport value (23/23; i.e., 1.0)
  Erosion is 7 tonnes/ha per year, 7
  Surface runoff class is very high, 8
  Field is connected, 8
Site management (145)
  Soil test P is 200, 40
  Fertilizer P application is 30 kg P/ha, 30
  Manure P application is 75 kg P/ha, 75

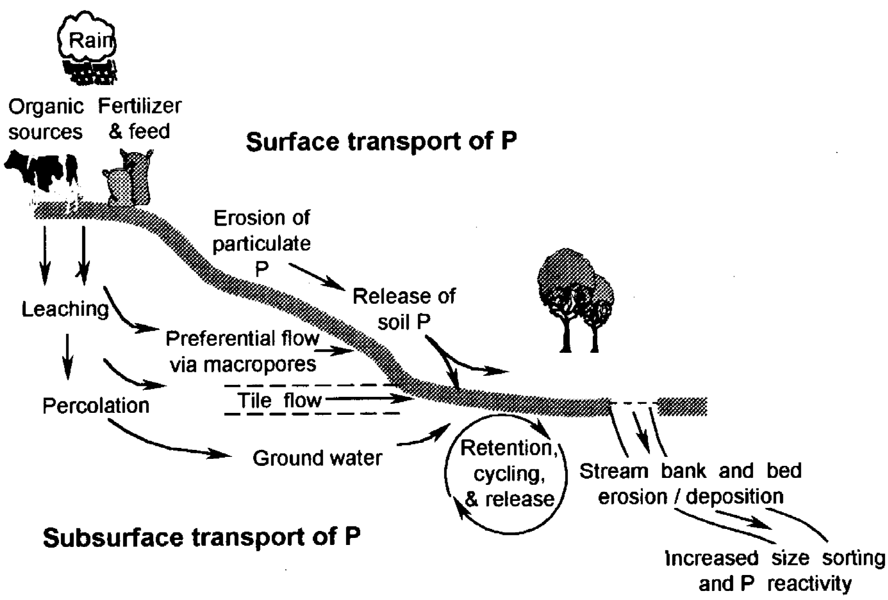

Figure 1. Transport and site management factors influencing the potential for P loss from agricultural land to surface waters.

Table 6. Management options to minimize nonpoint source pollution of surface waters by soil P

Phosphorus index	Management options to minimize nonpoint source pollution of surface waters by soil P
(LOW) < 30	*Soil testing*: have soils tested for P at least every 3 years to monitor build-up or decline in soil P. *Soil conservation*: follow good soil conservation practices. Consider effects of changes in tillage practices or land use on potential for increased transport of P from site.
(MEDIUM) 30–70	*Nutrient management*: consider effects of any major changes in agricultural practices on P losses *before* implementing them on the farm. Examples include increasing the number of animal units on a farm or changing to crops with a high demand for fertilizer P. *Soil testing*: have soils tested for P at least every 3 years to monitor build-up or decline in soil P. Conduct a more comprehensive soil testing program in areas that have been identified by the P Index as being most sensitive to P loss by surface runoff, subsurface flow, and erosion. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). *Nutrient management*: any changes in agricultural practices may affect P loss; carefully consider the sensitivity of fields to P loss before implementing any activity that will increase soil P. Avoid broadcast applications of P fertilizers and apply manures only to fields with lower P Index values.
(HIGH) 70–100	*Soil testing*: a comprehensive soil testing program should be conducted on the entire farm to determine fields that are most suitable for further additions of P. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). Consider using crops with high P removal capacities in fields with high P Index values. *Nutrient management*: in most situations fertilizer P, other than a small amount used in starter fertilizers, will not be needed. Manure may be in excess on the farm and should only be applied to fields with lower P Index values. A long-term P management plan should be considered.
(VERY HIGH) > 100	*Soil testing*: a comprehensive soil testing program must be conducted on the entire farm to determine fields that are most suitable for further additions of P. *Soil conservation*: implement practices to reduce P losses by surface runoff, subsurface flow, and erosion in the most sensitive fields (i.e., reduced tillage, field borders, grassed waterways, and improved irrigation and drainage management). Consider using crops with high P removal capacities in fields with high P Index values. *Nutrient management*: fertilizer and manure P should not be applied for at least 3 years and perhaps longer. A comprehensive, long-term P management plan must be developed and implemented.

P index is an educational tool that brings interaction between the planner and farmer in assessing environmental management decisions required to improve the farming system on a watershed rather than political basis.

Transport factors

Transport factors are critical to site assessment as they translate potential P sources into actual loss from a field or watershed. Factors controlling the transport of P within agricultural watersheds are conceptualized in Fig. 1. The main controlling factors and those considered in the P index are erosion, surface runoff, subsurface flow, and distance or connectivity of the site to the stream channel. The justification for inclusion of each of these factors is given below.

Erosion

Erosion is a mechanism of P transport that preferentially removes finer-sized soil particles (Haygarth and Sharpley, 2000). As a result, the P content and reactivity of eroded material is usually greater than source

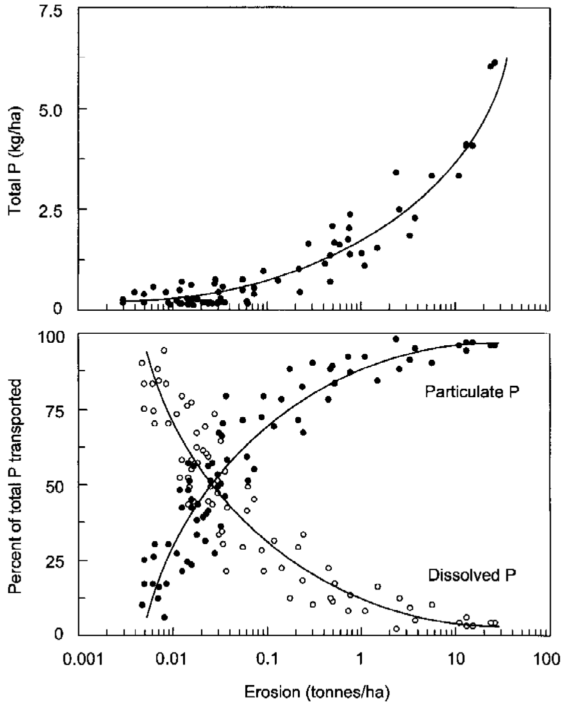

Figure 2. Total P loss and percentage of total P in dissolved and particulate forms a function of erosion in surface runoff from watersheds at E1 Reno, OK (adapted from Sharply et al., 1991; Smith et al., 1991).

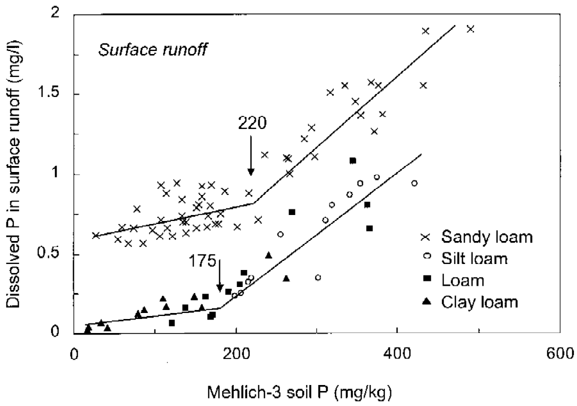

Figure 3. Relationship between the concentration of dissolved P in surface runoff and Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Co., PA (adapted form McDowell and Sharply, 2001).

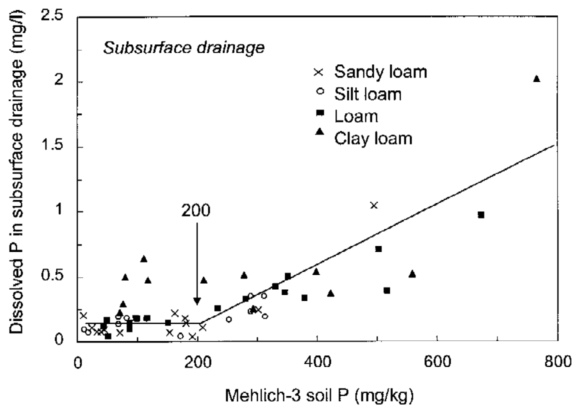

Figure 4. Relationship between the concentration of dissolved P in subsurface drainage from 30-cm deep lysimeters and the Mehlich-3 extractable soil P concentration of surface soil (0–5 cm) from the FD-36 watershed, Northumberland Cp., PA (adapted from McDowell and Sharply, 2001).

soil. For example, Sharpley (1985b) found that under simulated rainfall, the enrichment of soil test P (Bray-1 P) and total P content of sediment in surface runoff from several soils compared to the whole soil, ranged from 1.2 to 6.0 and 1.2 to 2.5, respectively. These P enrichment ratios increased as erosion decreased, favoring the relative movement of fine-particles (<2 μm) with greater P content over coarse particles (> 5 μm) with lower P content.

The effect of erosion on P movement is illustrated by a 15-year study of runoff from several grassed and cropped watersheds in the Southern Plains (Fig. 2; Sharpley et al., 1991; Smith et al., 1991). Increasing erosion from native grass, no-till and conventional-till wheat (*Triticum aestivum* L.) resulted in an increase in total P loss, of which a greater proportion was transported as particulate P. Accompanying the increase in particulate P movement, was a relative decrease in dissolved P movement (Fig. 2).

Surface Runoff

The potential for P loss in surface runoff from a given site can be extremely high. The transport of dissolved P in runoff is initiated by the release of P from soil, plant material, and suspended sediments (Fig. 1). This process occurs when rainfall interacts with a thin layer of surface soil (1–5 cm) before leaving the field as surface runoff (Sharpley, 1985a). The proportion of rainfall and depth of soil involved are highly dynamic due to variations in rainfall intensity, soil tilth, and vegetative cover, making them difficult to quantify in the field.

Subsurface Flow

Generally the P concentration in water percolating through the soil profile by leaching is small due to sorption of P by P-deficient subsoils. Exceptions occur in organic soils, where the adsorption affinity and capacity for P sorption are low due to the predominance of negatively charged surfaces (Duxbury and Peverly, 1978; Miller, 1979; White and Thomas, 1981). Other soils that are susceptible to movement include sandy soils with low P sorption capacities, waterlogged soils where Fe(III) has been reduced to Fe(II), and well structured soils prone to preferential flow through macropores and earthworm burrows (Bengston et al., 1992; Sharpley and Syers, 1979; Sims et al., 1998).

Because of the variable paths and time of water flow through a soil with subsurface drainage, factors controlling P loss in subsurface flow are more complex than for surface runoff. Subsurface flow includes artificial and natural drainage, where artificial drainage includes percolating water intercepted by installed drainage systems, such as mole and tile drains (Fig. 1). In general, the greater contact time between subsoil and natural subsurface flow than artificial drainage, results in lower losses of dissolved P in natural subsurface flow (Sharpley and Rekolainen, 1997; Sims et al., 1998).

Distance or connectivity to the stream channel

In order to translate the potential for P transport in surface runoff and subsurface flow from a given site to the potential for P loss in stream flow, it is necessary to account for whether water leaving a site actually reaches the stream channel. For instance surface runoff and subsurface flow may occur at various locations in a watershed and not reach the stream channel (Gburek et al., 2000). Thus, the location of a field in relation to the stream channel may determine whether runoff from the field reaches the channel and actually leaves the watershed. For a simple assessment of this factor, a site can be categorized as either not connected to the stream channel or connected to the channel by direct runoff, drainage ditch, or similar topographic feature.

Site Management Factors

A number of site management factors control P loss from agricultural lands. These include soil test P concentration, as well as rate, type (mineral fertilizer or manure), and method of P application (Fig. 1). These factors reflect day-to-day farm operations, while the transport factors discussed earlier tend to represent inherent soil, topographic and climatic properties.

Soil phosphorus

The loss of dissolved P in surface runoff is highly dependent on the P content of surface soil, as illustrated in Fig. 3. These data were obtained from several locations within a 40 ha watershed (FD-36) in south-central Pennsylvania (Northumberland Co.) using a portable rainfall simulator (Miller, 1987), following a protocol developed for the National P Project (Sharpley et al., 1999b). Briefly, either field plots (1-m wide and 2-m long) or packed boxes of soil (15-cm wide and 1-m long) were subjected to a rainfall intensity of 7 cm/h to produce 30-min of surface runoff, and a P concentration was determined for the entire 30-min event. This intensity for 30 min has an approximate 5-year return frequency in south-central Pennsylvania. To assess the role of soil test P on surface runoff P concentrations, field soils were selected to give a wide range in Mehlich-3 P concentrations (from 15 to 500 mg/kg).

A change point in the relationship between soil and surface runoff P, representing the interception of significantly different regression slopes ($P < 0.05$), is clearly visible at Mehlich-3 P values of 220 mg/kg (sandy loam soil) and 175 mg/kg (silt loam, loam and clay loam soils) (Fig. 3). Notably, for each of the soils, the potential for soil P release above this change point is greater than below it (McDowell and Sharpley, 2001; McDowell et al., 2001).

In a review of earlier studies, Sharpley et al. (1996) found that the specific regression equations between soil P and surface runoff P vary with soil type and management. For instance, regression slopes were flatter for grass (4.1–7.0, mean 6.0) than for cultivated land (8.3–12.5, mean 10.5). However, regression slopes were too variable to allow the use of a single or average relationship between soil test P and runoff P for all soils under the same management, probably due to inherent variability between soils. This variability is supported by the findings of Pote et al. (1999), who reported significantly different regression equations for three Ultisols of differing texture ($p < 0.05$). Also, the variation in the relationships presented in Fig. 3, as well as the corresponding change points, illustrates the soil specific nature of soil P release to surface runoff. Factors which influence P release among soils, include the dominant forms of P in soil, texture, aggregate diffusion, degree of interaction between soil and water, organic matter content, vegetative soil cover, and sorption capacities (Sharpley, 1983, 1999).

The concentration of P in subsurface flow is also related to surface soil P. In an experiment examining leachate from 30-cm deep lysimeters taken from the FD-36 watershed and subjected to simulated rainfall as described above for the National P Project, McDowell and Sharpley (2001) found the concentration of dissolved P in subsurface flow from the lysimeter increased (0.07–2.02 mg/l) as the Mehlich-3 P concentration of surface soil increased (15 to 775 mg/kg; Fig. 4). These data manifest a change point that was similar to the change point identified for surface runoff. They concluded that the dependence of subsurface P transport on surface soil P is evidence of the importance of P in preferential flow pathways such as earthworm burrows and old root channels.

Other studies have found a similar relationship between surface soil P and P loss in subsurface flow. For example, Heckrath et al. (1995) found that soil test P (Olsen P) >60 mg/kg in the plow layer of a silt loam, caused the dissolved P concentration in tile drainage water to increase dramatically (0.15–2.75 mg/l). They postulated that this level, which is well above that needed by major crops for optimum yield (about 20 mg/kg; Ministry of Agriculture, Food and Fisheries, 1994), is a critical point above which the potential for P movement in land drains greatly increases. Similar studies suggest that soil P thresholds can vary threefold as a function of site hydrology, relative drainage volumes, and soil P release (desorption) characteristics (McDowell et al., 2001; Sharpley and Syers, 1979).

Application of phosphorus as mineral fertilizer or manure

The application of mineral fertilizer and manure to soil may dramatically increase P loss in surface runoff and subsurface flow. For example, 14 days after applying either 0, 50, or 100 kg P/ha in dairy manure to a Berks silt loam (Typic Dystrochrept) with a Mehlich-3 P content of 75 mg/kg, we applied artificial rainfall following the National P Project protocol (Sharpley et al., 1999b) and observed dissolved P concentrations in surface runoff (7 cm/h rainfall for 30 min) of 0.25, 1.35, and 2.42 mg/l, respectively (Fig. 5).

Table 7 summarizes findings from a variety of studies examining the effect of mineral fertilizer and manure management on runoff P concentration. From this and earlier reviews (Sharpley and Rekolainen, 1997), it is clear that the loss of P is influenced by the rate, time, and method of application; form of P

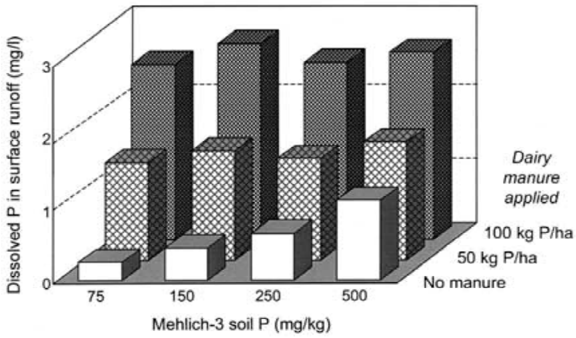

Figure 5. The concentration of P in surface runoff from a grassed Berks silt loam, as a function of Mehlich-3 soil P concentration and amount of dairy manure applied 2 weeks before the rainfall.

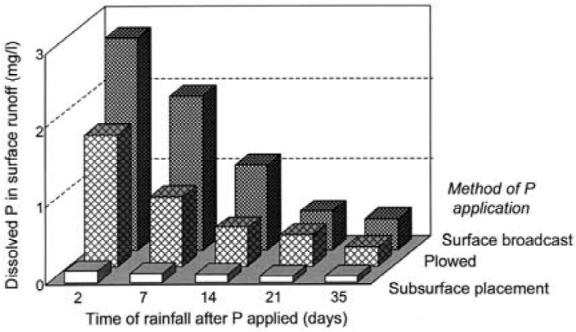

Figure 6. The effect of application method and timing of rainfall after application of dairy manure (100 kg p?ha) on the concentration of P in surface runoff from a grassed Berks silt loam.

added, amount and time of rainfall after application; and vegetative cover. In addition, the portion of applied P transported in runoff appears to be greater from conventional- than conservation-tilled watersheds. In one instance, McDowell and McGregor (1984) found mineral fertilizer P application to no-till corn actually reduced P transport, probably due to increased vegetative cover afforded by fertilization. Similarly, others found manure applications can reduce P loss in runoff via improved soil structure, aeration and water holding capacity afforded by added organic matter, as well as reducing erosion via increased vegetative over (Pote et al., 1996; Sharpley et al., 1998c).

Table 7 also illustrates that the loss of applied P in subsurface artificial drainage is appreciably lower than in surface runoff. Although it is difficult to distinguish between losses of mineral fertilizer, manure, or native soil P, without the use of expensive and hazardous radioactive tracers, total losses of applied P in runoff are generally less than 10% of that applied, unless rainfall immediately follows application or where surface runoff has occurred on steeply sloping, poorly drained,

Table 7. Effect of mineral fertilizer and manure application on P loss in surface runoff and fertilizer application on P loss in tile drainage

Land use	P added (kg ha^{-1} yr^{-1})	Phosphorus loss (kg ha^{-1} year^{-1})		Percent applied [a]	Reference and location
		Dissolved	Total		
Surface runoff					
Mineral Fertilizer					
Grass	0	0.02	0.22		McColl et al., 1977;
	75	0.04	0.33	0.1	New Zealand
No-till corn	0	0.70	2.00		McDowell and McGregor, 1984;
	30	0.80	1.80		Mississippi
Conventional corn	0	0.10	13.89		
	30	0.20	17.70	12.7	
Wheat	0	0.20	1.60		Nicolaichuk and Read, 1978;
	54	1.20	4.10	4.6	Saskatchewan, Canada
Grass	0	0.50	1.17		Sharpley and Syers, 1976;
	50	2.80	5.54	8.7	New Zealand
Grass	0	0.17	0.23		Uhlen, 1988;
	24	0.25	0.31	1.2	Norway
	48	0.42	0.49	1.0	
Dairy Manure [b]					
Alfalfa	0	0.10	0.10		Young and Mutchler, 1976;
-spring	21	1.90	3.70	17.1	Minnesota
-autumn	55	4.80	7.40	13.3	
Corn	0	0.20	0.10		
-spring	21	0.20	0.60	2.4	
-autumn	55	1.00	1.60	4.7	
Poultry Manure					
Grass	0	0.00	0.10		Edwards and Daniel, 1992;
	76	1.10	2.10	2.6	Arkansas
Grass	0	0.10	0.40		Westerman et al., 1983;
	95	1.40	12.4	12.6	North Carolina
Swine Manure					
Fescue	0	0.10	0.10		Edwards and Daniel, 1993a;
	19	1.50	1.50	7.4	Arkansas
	38	4.80	3.30	8.4	
Aritifical Drainage					
Corn	0	0.13	0.42		Culley et al., 1983;
	30	0.20	0.62	0.7	Ontario, Canada
Oats	0	0.10	0.29		
	30	0.20	0.50	0.7	
Potatoes + Wheat + Barley					Catt et al., 1997;
Minimal till	102	0.26	8.97	8.8	Woburn, England
Conventional till	102	0.35	14.38	14.1	
Alfalfa	0	0.12	0.32		
	30	0.20	0.51	0.6	
Grass - 0–30 cm	32	0.12	0.38	1.1	Heathwaite et al., 1997;
- 30–80 cm	32	0.76	1.77	5.5	Devon, UK
Grass	0	0.08	0.17		Sharpley and Syers, 1979;
	50	0.44	0.81	1.3	New Zealand

[a] Percent P applied lost in runoff.
[b] Manure applied in either spring or autumn.

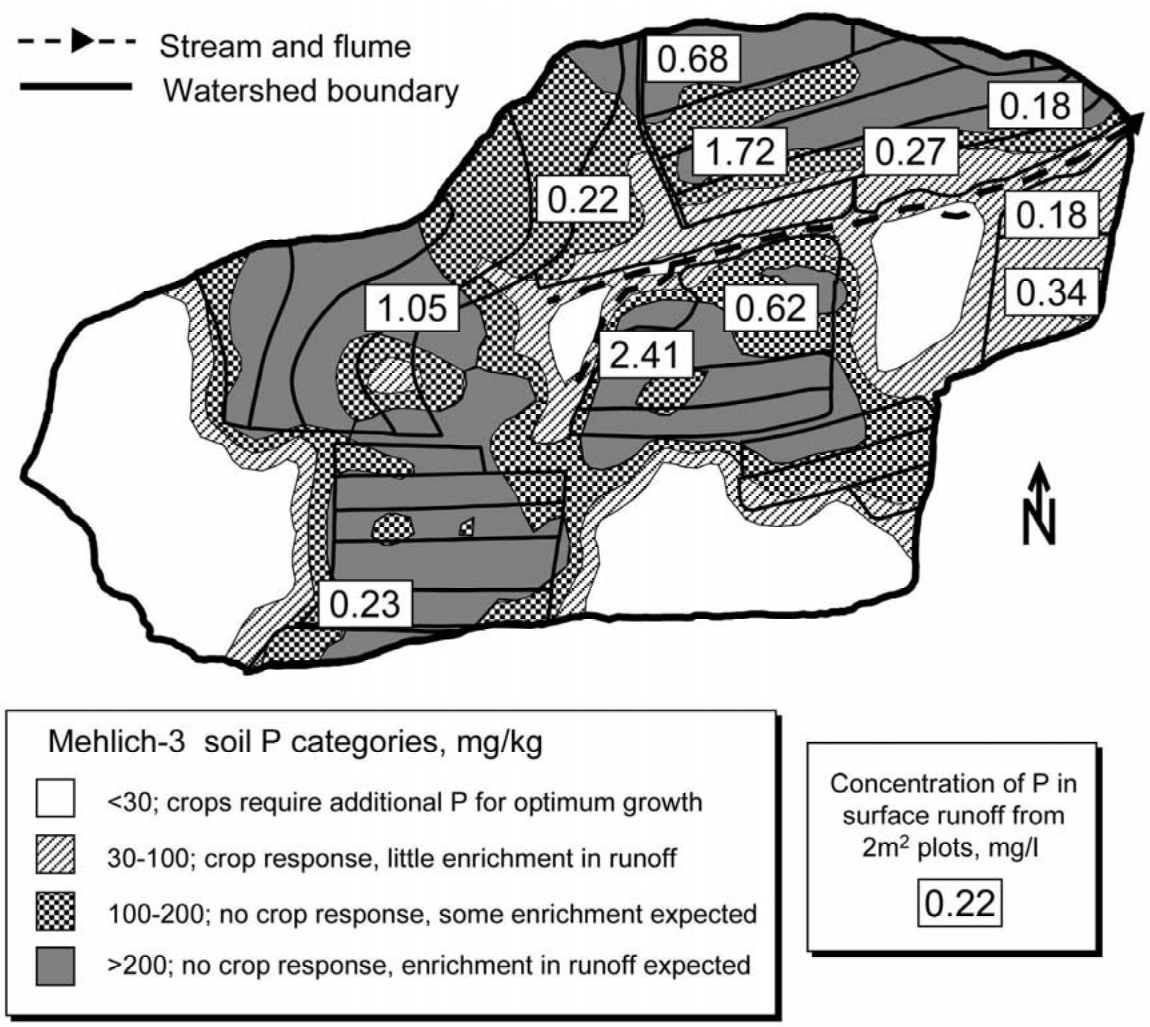

Figure 7. Distribution of Mehlich-3 soil P (0–5 cm soil depth) and concentration of dissolved P in surface runoff from 2-m^2 plots within the FD-36 watershed, Northumberland Co., PA.

and/or frozen soils. The high proportion of manurial P in runoff reported by the studies summarized in Table 7 may result from high manure application and generally less flexibility in application timing than for mineral fertilizer. Such inflexibility in scheduling of manure application results from the continuous production of manure throughout the year and a frequent lack of manure storage facilities.

Although we have shown soil P is important in determining P loss in surface runoff, applying P to soil can override soil P in determining P loss. For example, in our simulated rainfall study in the FD-36 watershed, the dissolved P concentration of surface runoff increased with Mehlich-3 P concentration in the upper 5 cm of soil (Fig. 5). When dairy manure was broadcast on these grassed soils, the dissolved P concentration of surface runoff 14 days later, was greater than with no manure (Fig. 5). Furthermore, the application of increasing quantities of manure P to these soils masked the effect of soil P concentration on surface runoff P.

Phosphorus application method and timing relative to rainfall also influences the concentration of P removed in runoff. For example, several studies have shown a decrease in P loss with an increase in the length of time between P application and surface runoff (Edwards and Daniel, 1993b; Sharpley, 1997; Westerman et al., 1983). This decrease can be attributed to the reaction of added P with soil and dilution of applied P by infiltrating water from rainfall that did not cause surface runoff. For instance, in our rainfall simulation studies in the FD-36 watershed, the dissolved P concentration of surface runoff from the Berks silt loam decreased from 2.75 to 0.40 mg/l when rainfall occurred 35 days rather than 2 days after a surface broadcast application of 100 kg P/ha as dairy manure (Fig. 6).

Although the concentration of P at the soil surface serves as the primary source of P to runoff, incorporation of manure into the soil profile either by tillage or subsurface placement, decreases the potential for P loss in surface runoff (Fig. 6). For example, the dissolved P concentration of surface runoff from a Berks silt loam 2 days after the surface application of 100 kg P/ha dairy manure was 2.75 mg/l. When the same amount of manure was incorporated by plowing to a depth of 10 cm, dissolved P in surface runoff was 1.70 mg/l, and when the manure was placed 5 cm below the soil surface, dissolved P in surface runoff fell to 0.15 mg/l (Fig. 6).

In an earlier field study, Mueller et al. (1984) found that incorporation of dairy manure by chisel plowing reduced total P loss in runoff from corn 20-fold compared to no-till areas receiving surface applications. However, the concentration of P in surface runoff did not decrease as dramatically as the mass of P lost. This was due to an increase in infiltration rate with manure incorporation and consequent decrease in surface runoff volume. In fact, surface runoff volume from no-till corn was greater than from conventional-till corn. Thus, P loss in runoff decreased by a dilution of P at the soil surface and reduction in runoff with incorporation of manure.

Testing the P index

Although there is a great deal of research documenting the justification of the transport and source factors included in the P index, there has been little site evaluation of index ratings. The original and modified versions of the P index have been used to assess the potential for P loss in several regions including the Delmarva Peninsula (Leytem et al., 1999; Sims, 1996), Oklahoma (Sharpley, 1995), Texas (McFarland et al., 1998), Vermont (Jokela et al., 1997), and Canada (Bolinder et al., 1998). However, few comparisons of P index ratings and measured P loss have been made. In Nebraska, Eghball and Gilley (1999) found correlation coefficients (r) as high as 0.84, between total P loss from simulated rainfall-runoff plots and P index ratings, when erosion losses were strongly weighted in the P index.

Using the National P Project rainfall simulator (Sharpley et al., 1999b), we measured the dissolved P concentration in surface runoff from 48, 1×2-m plots within the FD-36 watershed to evaluate the ability of the P index to rank site vulnerability to P loss on a plot scale. A selection of dissolved P concentrations of surface runoff within the watershed is given in Fig. 7, along with surface soil (0–5 cm depth) Mehlich-3 P illustrating the large variation in surface runoff P concentrations found between plots. At some sites, rain simulation was conducted approximately 2 weeks after manure application. At other sites, no manure had been applied for at least 9 months prior to rain simulation. Thus, the range in dissolved P concentration was a function of soil P concentration and manure application.

The P index was applied to each plot within the FD-36 watershed. Using soil survey, land management, and topographic information, erosion was calculated by the Revised Universal Soil Loss Equation (RUSLE) and surface runoff by the curve number approach (Sharpley et al., 1998a). Site management factors of the P index were calculated from Mehlich-3 P concentration of surface soil (0–5 cm depth) and P application rate, method, and timing as shown in Table 4. The final P index rating for each plot was calculated as the product of transport and site management factors as described in Table 5. Due to our use of plot data, the evaluation of the P index did not account for landscape factors such as site position or connectivity, which precludes the interpretation of P index rating values by the management categories given in Tables 5 and 6.

Figure 8 illustrates the relationship between plot P index ratings and dissolved P in surface runoff. The two variables were strongly associated (r^2 = 0.78; P = 0.001). This strong association indicates the P index can accurately account for and describe a site's potential for P loss if surface runoff were to occur (Fig. 8).

In addition to this plot-scale assessment of the P index, a watershed-scale validation is required of the index, leaving a number of questions that must be addressed. For instance, are the areas identified to be at greatest risk for P loss, actually sources of most of the P exported? In the same vein, will remediation of high risk areas identified by the index decrease P export in stream flow from a watershed? Conversely, can low

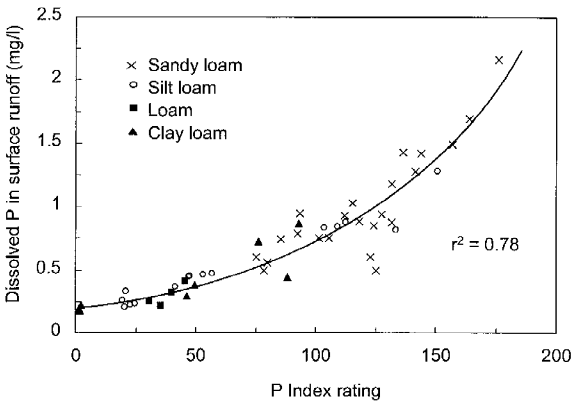

Figure 8. Relationship between the concentration of dissolved P in surface runoff and P index rating 2-m^2 plots within the FD-36 watershed, Northumberland Co., PA.

vulnerability areas receive more liberal P management without increasing P export?

Finally, and perhaps most critical to the use of the P index to as a guide to P management practices, will be the development of overall risk assessment classifications (see Tables 5 and 6). These classifications and interpretations must be developed with careful consideration of local management options, industry infrastructures, and State and Federal policy programs. With further development and testing, the P index will be a valuable tool to identify critical areas of P export, so that alternative management options and remedial measures can be identified. Limited resources and assistance can then be better used to target remedial measures to areas where they will have the most benefit.

Remedial measures

Remedial measures must begin with the long-term objective of increasing P use-efficiency, by attempting to balance P inputs within a watershed with P outputs, while simultaneously improving management of soil, manure and mineral fertilizer P. Reducing P loss in agricultural runoff may be brought about by source and transport control strategies, such as those listed in Table 8. In the past, much attention has been focused on erosion control as a means of controlling P loss from agricultural land. Increasingly, however, attention is being directed toward source management and the control of dissolved P losses in surface runoff.

Source management

Source management attempts to minimize the buildup of P in the soil above levels sufficient for optimum crop growth, by limiting the quantity of P in manure that must be applied to land, and controlling the amount of P that is applied in a localized area. Techniques for source management include:

- Manipulation of dietary P intake by animals may help reduce P inputs in feed; often the major cause of P surplus. Phosphorus intake in excess of minimum dietary requirements do not appear to confer any growth or health advantages and actually decreases profitability through increased feed costs (Knowlton and Kohn, 1999). Carefully matching dietary P inputs to livestock requirements can reduce the amounts of P excreted by animals.
- Increasing the efficiency of P uptake by livestock from feed. A significant amount of the P in grain is in phytate (phytic acid), a form of P that cannot be digested by monogastric animals such as pigs and chickens. As a result, it is common to supplement feed with mineral forms of P, which contribute to P enrichment of manures and litters. Enzymes such as phytase, which break down phytate into forms available to monogastric animals, can be added to feed to increase the efficiency of grain P absorption by pigs and poultry. Such enzymes reduce the need for P supplements in feed and potentially reduce the P content of manure. Also, corn hybrids are available which contain low amounts of indigestible phytate P. Pigs and chickens fed 'low-phytic acid' corn grain excreted less P in manure than those fed conventional corn varieties (Ertl et al., 1998).
- Use of manure and soil testing data to improve nutrient management. Farm advisors and resource planners are now recommending that the P content of both manure and soil be determined by soil test laboratories before land application of manure.
- Use of amendments to decrease P solubility in soil and manure. Commercially available manure amendments, such as slaked lime ($CaOH_2$) or alum ($Al_2(SO_4)_3$), are used to reduce ammonia (NH_3) volatilization, leading to improved animal health and weight gains. Coincidentally, these amendments can also greatly decrease the water solubility of P in poultry litter, thereby decreasing dissolved P concentrations in surface runoff (Moore et al., 2000; Shreve et al., 1995). Perhaps the most important benefit of manure amendments

Table 8. Best Management Practices for control of nonpoint sources of agricultural P and N

Practice	Description	Impact on loss[a]	
		P	N
Source Measures			
Feed additives	Enzymes increase nutrient utilization by animals	Decrease	Decrease
Crop hybrids	Low phytic-acid corn reduces P in manure	Decrease	Neutral
Manure management	Compost, lagoons, pond storage; barnyard runoff control; transport excess out of watershed	Decrease	Decrease
Rate added	Match crop needs	Decrease	Decrease
Timing of application	Avoid autumn and winter application	Decrease	Decrease
Method of application	Incorporated, banded, or injected in soil	Decrease	Decrease
Crop rotation	Sequence different rooting depths	Neutral	Decrease
Manure amendment	Alum reduces NH_3 loss and P solubility	Decrease	Decrease
Soil amendment	Flyash, Fe oxides, gypsum reduce P solubility	Decrease	Neutral
Cover crops/residues	If harvested can reduce residual soil nutrients	Decrease TP	Increase DP
Plowing stratified soils	Redistribution of surface P through profile	Decrease	Neutral
Transport Measures			
Cultivation timing	Not having soil bare during winter	Decrease	Decrease
Conservation tillage	Reduced and no-till increases infiltration and reduces soil erosion	Decrease TP Increase DP	Decrease Increase NO_3
Grazing management	Stream exclusion, avoid overstocking	Decrease	Decrease
Buffer, riparian, wetland areas, grassed waterways	Removes sediment-bound nutrients, enhances denitrification	Decrease TP neutral DP	Decrease
Soil drainage	Tiles and ditches enhance water removal and reduce erosion	Decrease TP Increase DP	Decrease TN Increase NO_3
Strip cropping, contour plowing, terraces	Reduces transport of sediment-bound nutrients	Decrease Neutral DP	Decrease Neutral NO_3
Sediment delivery structures	Stream bank protection and stabilization, sedimentation pond	Decrease	Decrease
Critical source area treatment	Target sources of nutrients in a watershed for remediation	Decrease	Decrease

[a]TN is total N, NO_3 is nitrate, TP is total P, and DP is dissolved P.

for both air and water quality would be an increase in the N:P ratio of manure (via reduced N loss because of NH_3 volatilization) that would more closely match crop N and P requirements.

- Transporting manure P from areas of P excess to areas of P deficiency. At present, manure is rarely transported more than 15 km from where it is produced, restricting application options. However, it must be shown that the recipient farms are more suitable for manure application than manure-rich farms and that measures are managed on recipient farms to avoid soil P build up.

- Composting may also be considered as a management tool to improve manure distribution. Although composting tends to increase the P concentration of manure, the volume is reduced and thus, transportation costs are reduced.

- Separating solids from liquids may increase the number of management options available for some types of manure such as dairy and swine. This process results in some separation of the nutrients as well, leaving a large proportion of the available N in the liquid fraction and a large proportion of the P will be in the solid fraction. While this does not change the total amount of nutrients that must be handled, it may enable better targeting of the individual nutrients to locations where they will do the most good and/or have less potential for causing environmental problems. Also, because the solid fraction is more concentrated it may be feasible to transport it to more remote fields.

- Using manure as a source of 'bioenergy'. For example, dried poultry litter can be burned directly or converted by pyrolytic methods into oils suitable for use to generate electric power. Liquid manures can be digested anaerobically to produce methane which can be used for heat and energy.
- Improving management of P application rate, method, and timing to minimize the potential for P loss in runoff. As we have shown, P loss in runoff increases with greater applications of P as mineral fertilizer or manure (Table 7 and Figs. 5 and 6). Incorporation of manure into the soil profile either by tillage or subsurface placement, decreases the potential for P loss in runoff by lowering the concentration of P at the soil surface and a reducing runoff volume (Mueller et al., 1984; Pote et al., 1996).

Transport management

Transport management refers to efforts to control the movement of P from soils to sensitive locations such as bodies of fresh water. Phosphorus loss via surface runoff and erosion may be reduced by conservation tillage and crop residue management, buffer strips, riparian zones, terracing, contour tillage, cover crops and impoundments (e.g., settling basins). Basically, these practices reduce rainfall impact on the soil surface, reduce surface runoff volume and velocity, and increase soil resistance to erosion. Conversion from furrow irrigation to sprinkler to drip irrigation significantly reduces irrigation erosion and runoff. Furrow treatments such as straw mulching and use of polyacrylamides will also reduce in-furrow soil movement (Lentz et al., 1998).

Despite these advantages, any one of these measures should not be relied upon as the sole or primary means of reducing P losses. These practices are generally more efficient at reducing sediment P than dissolved P. Also, P stored in stream and lake sediments can provide a long-term source of P in waters long after inputs from agriculture have been reduced. Several researchers have indicated little decrease in lake productivity with reduced P inputs following implementation of conservation measures (Gray and Kirkland, 1986; Young and DePinto, 1982). Thus, the effect of remedial measures in the contributing watershed will be slow for many cases of poor water quality. Therefore, immediate action may be needed to reduce future problems.

Integrating P and N management

Farm N inputs are usually more easily balanced with plant uptake than are P inputs, particularly where confined livestock operations exist. In the past, separate strategies for either N or P have been developed and implemented at farm or watershed scales. Because of different critical sources, pathways, and sinks controlling P and N export from watersheds, remedial efforts directed at either P or N control can negatively impact the other nutrient (Table 8). For example, basing manure application on crop N requirements to minimize nitrate leaching to ground water can increase soil P and enhance potential P losses (Sharpley et al., 1998b; Sims, 1997). In contrast, reducing surface runoff losses of total P via conservation tillage can enhance N leaching and even increase algal available P transport (Sharpley and Smith, 1994).

These positive and negative impacts of conservation practices on N and P loss potential should be considered in the development of sound remedial measures. Clearly, a technically sound framework must be developed that recognizes critical sources of P and N export from agricultural watersheds so that optimal strategies at farm and watersheds scales can be implemented to best manage both P and N. One approach, explored by Heathwaite et al. (2000) and Sharpley et al. (1998a), is to employ the P index to target P management on critical source areas of P and assume N-based management on all other areas. With such an approach, however, careful consideration must be given to the potential long-term consequences of N management on P loss and vice versa.

Bridging agricultural and environmental management

In order to initiate real and lasting changes in agricultural production, emphasis must be placed on consumer-based programs and education rather than assuming that farmers will absorb the burden. Acceptance of best management practices (BMPs) will not be easy. Because farmers' decisions are generally shaped by regional and often global economic pressures and constraints, which they have little or no control over, there is often reluctance to adopt management practices that do not address these concerns. Clearly, new ways of using incentives to help farmers implement BMPs are needed. The challenge is to recognize how social policy and economic factors influence the nutrient-management agenda.

Equally important is that everyone is affected by and can contribute to a resolution of nutrient-related concerns. Rather than assume that inappropriate farm management is responsible for today's water quality problems, the underlying causes of the symptoms must be addressed. As shown above, much of today's problems relate to marketplace pressures, the breakdown and imbalances in global P cycling, and economic survival of farms. Research is, thus, needed to develop programs that encourage farmer performance and stewardship to achieve previously agreed upon environmental goals. These programs should focus on public participation to resolve conflicts between economic production efficiency and water quality. In the US, there are numerous sources of technical assistance and financial cost-share and loan programs to help defray the costs of constructing or implementing practices that safeguard soil and water resources (US Environmental Protection Agency, 1998). Watershed-based programs, such as the New York City Watershed Agriculture Program, have been established to provide technical assistance and financial support to farmers participating in water quality protection programs (National Research Council, 2000).

Stakeholder alliances encourage collaborative relationships among concerned parties. Such alliances have been formed in response to recent public health issues related to the nutrient enrichment of waters in the eastern US. In the Chesapeake Bay, stakeholder alliances have developed among state, federal, and local groups and the public to work together to identify critical problems, focus resources, include watershed goals in planning, and implement effective strategies to safeguard soil and water resources (Chesapeake Bay Program, 1995, 1998).

In Australia, it was found that an awareness of agricultural or environmental problems and potential solutions did not necessarily cause people to change their behavior to correct such problems (Wilkinson and Cary, 1993). Solutions have to be adapted in practical ways to individual circumstances. The Australian National Land and Water Resources Audit has recognised this by investigating the capacity of rural communities to implement changes to help protect soil and water (National Land and Water Resources Audit, 1998).

One barrier to the design and implementation of BMPs is that the assessment and monitoring implemented by government is often perceived as a top-down process. Walker et al. (1996) recommend a bottom-up process, whereby policy maker and user can select soil and water quality indicators at a local level. Such a process would seem equally if not more fundamental to the successful identification and adoption of new management systems. A concerted attempt has been made in Australia to take this approach by devolving primary responsibility for local monitoring and resource management to land managers themselves through the provision of government funds for the national Landcare and Waterwatch programs (SCARM-ARMCANZ, 1997).

Finally, P applications at recommended rates can reduce P loss in agricultural runoff via increased crop uptake and cover. It is of vital importance that we implement management practices that minimize soil P buildup in excess of crop requirements, reduce surface runoff and erosion, and improve our capability to identify fields that are major sources of P loss to surface waters.

Summary

A growing focus on nutrient transfers from agricultural lands to water has served to accelerate our understanding of the environmental consequences of P management in agriculture. Phosphorus imbalances at farm and watershed scales, often related to concentrated animal feeding operations, aggravate diffuse P losses through the gradual accumulation of P in soils, and the application of P at times of high transport potential. While erosion of particulate P from agricultural soils remains a dominant concern, the transport of dissolved, or soluble P in surface runoff and subsurface flow is also important.

Research at plot, field and watershed scales emphasizes the importance of critical source areas, where high P availability and high transport potential overlap, as major contributors to P losses from agricultural lands to water. The development of tools such as the P index represents a major advance in identifying critical source areas, as highlighted by our research relating P index ratings to plot-scale P losses.

Management strategies to minimize P loss to water may be brought about by optimizing P use-efficiency, refining animal feed rations, using feed additives to increase P absorption by the animal, moving manure from surplus to deficit areas, and targeting conservation practices, such as reduced tillage, buffer strips and cover crops, to critical source areas within a watershed. However, because farmers' decisions are influenced by regional and even global economics over which they have little control, we should explore the use of incentives to aid in implementation of innovative measures that minimize on-farm surpluses of P and reduce P losses.

References

Beegle D 2000 Soil fertility management. *In* Agronomy Guide 1999–2000. Eds. N Seroktin and S Tibbetts. pp 19–46. University Park, PA: Publications Distributions Center, Pennsylvania State University.

Bengston L, Seuna P, Lepisto A and Saxena R K 1992 Particle movement of meltwater in a subdrained agricultural basin. J. Hydrol. 135, 383–398.

Bolinder M A, Simard R R, Beauchemin S and MacDonald K B 1998 Indicator of risk of water contamination: methodology for the phosphorus component. pp 11–21, Agri-environmental Indicator Project Report No. 24, Agriculture and Agri-food Canada.

Burkholder J A and Glasgow H B Jr 1997 *Pfiesteria piscicidia* and other Pfiesteria-dinoflagellates behaviors, impacts, and environmental controls. Limnol. Oceanogr. 42, 1052–1075.

Carpenter S R, Caraco N F, Correll D L, Howarth R W, Sharpley A N and Smith V H 1998 Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8, 559–568.

Catt J A, Johnston A E and Quinton J N 1997 Phosphate losses in the Woburn erosion reference experiment. *In* Phosphorus Loss from Soil to Water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 374–377. CAB International, Wallingford, UK.

Chesapeake Bay Program 1995 *In* The State of the Chesapeake Bay 1995. Eds. R Magnien, D Boward, and S Bieber. U.S. Government Printing Office, Washington, DC. 45 p.

Chesapeake Bay Program 1998 Federal Agencies Chesapeake Ecosystem Unified Plan. Chesapeake Bay Program, Annapolis, MD.

Culley J L B, Bolton E F and Bernyk V 1983 Suspended solids and phosphorus loads from a clay soil: I. Plot studies. J. Environ. Qual. 12, 493–498.

Duxbury J M and Peverly J H 1978 Nitrogen and phosphorus losses from organic soils. J. Environ. Qual. 7, 566–570.

Edwards D R and Daniel T C 1992 Potential runoff quality effects of poultry manure slurry applied to fescue plots. Trans. ASAE 35, 1827–1832.

Edwards D R and Daniel T C 1993a Runoff quality impacts of swine manure applied to fescue plots. Trans Am. Soc. Agric. Eng. 36, 81–80.

Edwards D R and Daniel T C 1993b Drying interval effects on runoff from fescue plots receiving swine manure. Trans. Am. Soc. Agric. Eng. 36, 1673–1678.

Eghball B and Gilley J E 1999 Phosphorus risk assessment index and comparison with results of runoff studies. p 30. *In* Agronomy Abstracts 1999. Am. Soc. Agron., Madison, WI.

Ertl D S, Young K A and Raboy V 1998 Plant genetic approaches to phosphorus management in agricultural production. J. Environ. Qual. 27, 299–304.

Gburek W J and Sharpley A N 1998 Hydrologic controls on phosphorus loss from upland agricultural watersheds. J. Environ. Qual. 27, 267–277.

Gburek W J, Sharpley A N, Heathwaite A L, and Folmar G J 2000 Phosphorus management at the watershed scale: a modification of the phosphorus index. J. Environ. Qual. 29, 130–144.

Gray C B J and Kirkland R A 1986 Suspended sediment phosphorus composition in tributaries of the Okanagan Lakes, BC. Water Res. 20, 1193–1196.

Haygarth P M and Sharpley A N 2000 Terminology for phosphorus transfer. J. Environ. Qual. 29, 10–15.

Haygarth P M, Hepworth L and Jarvis S C 1998 Form of phosphorus transfer and hydrological pathways from soil under grazed grassland. Eur. J. Soil Sci. 49, 65–72.

Heathwaite A L, Griffiths P, Haygarth P M, Jarvis S C and Parkinson R J 1997 Phosphorus loss from grassland soils: implications of land management for the quality of receiving waters. pp 177-186. *In* Freshwater Contamination, Proc. Rabat Symp., April–May 1997, IHAS Publ. No 243.

Heathwaite A L, Sharpley A N and Gburek W J 2000 Integrating phosphorus and nitrogen management at catchment scales. J. Environ. Qual. 29, 158–166.

Heckrath G, Brookes P C, Poulton P R and Goulding K W T 1995 Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk experiment. J. Environ. Qual. 24, 904–910.

Jokela W, Magdoff F and Durieux R 1997 Soil testing for improved phosphorus management. Vermont Cooperative Extension Service, Burlington, VT.

Kleinman P J A, Bryant R B, Reid W S, Sharpley A N and Pimental D 2000. Using soil phosphorus behavior to identify environmental thresholds. Soil Sci.

Knowlton K F and Kohn R 1999 We've got to stop overfeeding phosphorus. Hoard's Dairyman 144, 437.

Kotak B G, Kenefick S L, Fritz D L, Rousseaux C G, Prepas E E and Hrudey S E 1993 Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. Water Res. 27, 495–506.

Lander C H, Moffitt D, and Alt K 1998 Nutrients Available from Livestock Manure Relative to Crop Growth Requirements. Resource Assessment and Strategic Planning Working Paper 98–1. US Department of Agriculture, Natural Resources Conservation Service, Washington, DC. (http://www.nhq.nrcs.usda.gov/land/pub s /nlweb.html).

Lanyon L E 2000 Nutrient management: Regional issues affecting the Chesapeake Bay. *In* Agriculture and Phosphorus Management: The Chesapeake Bay. Ed. A N Sharpley. CRC Press, Boca Raton, FL.

Lemunyon J L and Gilbert R G 1993 The concept and need for a phosphorus assessment tool. J. Prod. Agric. 6, 483–496.

Lentz R D, Sojka R E, and Robbins C W 1998 Reducing phosphorus losses from surface-irrigated fields: emerging polyacrylamide technology. J. Environ. Qual. 27, 305–312..

Leytem A B, Sims J T, Coale F J Sharpley A N and Gburek W J 1999 Implementing a phosphorus site index for the Delmarva Peninsula: challenges and research needs. pp. 336. *In*: Agronomy Abstracts 1999. Am. Soc. Agron., Madison, WI.

Lory J A and Scharf P C 2000 Threshold phosphorus survey. SERA-17 Minimizing phosphorus losses from agriculture. (http://ces.soil.ncsu.edu/sera17/).

McColl R H S, White E and Gibson A R 1977 Phosphorus and nitrate runoff in hill pasture and forest catchments, Taita, New Zealand. N.Z. J. Mar. Freshwater Res. 11, 729–744.

McDowell L L and McGregor K C 1984 Plant nutrient losses in runoff from conservation tillage corn. Soil Tillage Res. 4, 79–91.

McDowell R W and Sharpley A N 2001 Approximating phosphorus release to surface runoff and subsurface drainage. J. Environ. Qual. 30, 508–601.

McDowell R W, Sharpley A N, Brookes P A and Poulton P 2001 Relationships between soil test phosphorus and phosphorus release to solution. Soil Sci. (in press).

McFarland A, Hauck L, White J, Donham W, Lemunyon J and Jones S 1998 Nutrient management using a phosphorus risk index for manure application fields. pp 241–244. Proceedings of Manure Management in Harmony with the Environment and Society, Soil and Water Conservation Society, Feb. 10–12, 1998, Ames, IA.

Miller M H 1979 Contribution of nitrogen and phosphorus to subsurface drainage water from intensively cropped mineral and organic soils in Ontario. J. Environ. Qual. 8, 42–48.

Miller W P 1987 A solenoid-operated, variable intensity rainfall simulator. Soil Sci. Soc. Am. J. 51, 832–834.

Ministry of Agriculture, Food and Fisheries 1994 Fertilizer recommendations for agricultural and horticultural crops. Ministry of Agriculture, Fisheries, and Food, Reference Book 209. HMSO, London, England. 48 p.

Moore P A Jr, Daniel T C and Edwards D R 2000 Reducing phosphorus runoff and inhibiting ammonia loss from poultry manure with aluminum sulfate. J. Environ. Qual. 29, In press.

Mueller D H, Wendt R C and Daniel T C 1984 Phosphorus losses as affected by tillage and manure application. Soil Sci. Soc. Am. J. 48, 901–905.

National Land and Water Resources Audit 1998 National Land and Water Resources Audit: Draft Strategic Plan — 1998–2001. National Land and Water Resources Audit, Canberra, Australia.

National Research Council (NRC) 2000 Watershed Management for Potable Water Supply. National Academy Press, Washington, DC.

Nicholaichuk W and Read D W L 1978 Nutrient runoff from fertilized and unfertilized fields in western Canada. J. Environ. Qual. 7, 542–544.

Pionke H B, Gburek W J, Sharpley A N, and Zollweg J A 1997 Hydrologic and chemical controls on phosphorus losses from catchments. *In* Phosphorus Loss to Water from Agriculture. Eds. H Tunney, O Carton and P Brookes. pp 225–242. CAB International, Cambridge, England.

Pote D H, Daniel T C, Sharpley A N, Moore P A Jr, Edwards D R and Nichols D J 1996 Relating extractable phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am. J. 60, 855–859.

Pote D H, Daniel T C, Nichols D J, Sharpley A N, Moore, P A Jr, Miller D M, and Edwards D R 1999 Relationship between phosphorus levels in three Ultisols and phosphorus concentrations in runoff. J. Environ. Qual. 28, 170–175.

SCARM-ARMCANZ 1997 Evaluation Report on the Decade of Landcare Plan — National Overview, Australian Government Publishing Service, Canberra, Australia.

Schoumans O F and Breeuwsma A 1997 The relation between accumulation and leaching of phosphorus: laboratory, field and modelling results. *In* Phosphorus loss from soil to water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 361–363. CAB International Press, Cambridge, UK.

Sharpley A N 1983 Effect of soil properties on the kinetics of phosphorus desorption. Soil Sci. Soc. Am. J. 47: 462–467.

Sharpley A N 1985a Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. Soil Sci. Soc. Am. J. 49, 1010–1015.

Sharpley A N 1985b The selective erosion of plant nutrients in runoff. Soil Sci. Soc. Am. J. 49, 1527–1534.

Sharpley A N 1995 Identifying sites vulnerable to phosphorus loss in agricultural runoff. J. Environ. Qual. 24, 947–951.

Sharpley A N 1997 Rainfall frequency and nitrogen and phosphorus in runoff from soil amended with poultry litter. J. Environ. Qual. 26, 1127–1132.

Sharpley A N 1999 Phosphorus availability. Section D, pages 18–38. *In* M. Sumner, Handbook of Soil Science. CRC Press, Boca Raton, FL.

Sharpley A N 2000 Editor Agriculture and Phosphorus Management: The Chesapeake Bay. CRC Press, Boca Raton, FL.

Sharpley A N and Rekolainen S 1997 Phosphorus in agriculture and its environmental implications. *In* Phosphorus Loss from Soil to Water. Eds. H Tunney, O T Carton, P C Brookes and A E Johnston. pp 1–54. CAB International Press, Cambridge, UK.

Sharpley A N and Smith S J 1994 Wheat tillage and water quality in the Southern Plains. Soil Tillage Res. 30: 33–48.

Sharpley A N and Syers J K 1976 Phosphorus transport in surface runoff as influenced by fertilizer and grazing cattle. N.Z. J. Sci. 19, 277–282.

Sharpley A N and Syers J K 1979 Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. N.Z. J. Agric. Res. 22, 127–131.

Sharpley A N and Tunney H 2000 Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. J. Environ. Qual. 29, 176–181.

Sharpley A N, Daniel T C, Sims J T and Pote D H 1996 Determining environmentally sound soil phosphorus levels. J. Soil Water Conserv. 51, 160–166.

Sharpley A N, Daniel T C , Sims J T, Lemunyon J, Steven R A and Parry R 1999a Agricultural phosphorus and eutrophication. US Department of Agriculture — Agricultural Research Service, ARS-149 U.S. Govt. Printing Office, Washington, DC 34 p.

Sharpley A N, Gburek W J and Folmar G 1998a Integrated phosphorus and nitrogen management in animal feeding operations for water quality protection. *In* Animal Feeding Operations and Ground Water: Issues, Impacts, and Solutions. Eds. R W Masters and D Goldman. pp. 72–95. National Ground Water Association, Westerville, OH.

Sharpley A N, Gburek W J and Heathwaite A L 1998b Agricultural phosphorus and water quality: Sources, transport and management. J. Agric. Food Chem. Finland 7, 297–314.

Sharpley A N, Smith S J, Williams J R, Jones O R and Coleman G A 1991 Water quality impacts associated with sorghum culture in the Southern Plains. J. Environ. Qual. 20, 239–244.

Sharpley A, Daniel T, Wright B, Kleinman P, Sobecki T, Parry R and Joern B 1999b National research project to identify sources of agricultural phosphorus loss. Better Crops 83, 12–15.

Sharpley A N, Meisinger J J, Breeuwsma A., Sims T, Daniel T C and Schepers J S 1999c Impacts of animal manure management on ground and surface water quality. p. 173–242. *In* J. Hatfield, Effective Management of Animal Waste as a Soil Resource. Ann Arbor Press, Chelsea, MI.

Shreve B R, Moore P A Jr, Daniel T C, Edwards D R and Miller D M 1995 Reduction of phosphorus in runoff from field-applied poultry litter using chemical amendments. J. Environ. Qual. 24, 106–111.

Sims J T 1996 The Phosphorus Index: a phosphorus management strategy for Delaware's agricultural soils. Fact Sheet ST-05, Delaware Cooperative Extension Service, Newark. DE.

Sims J T 1997 Agricultural and environmental issues in the management of poultry wastes: recent innovations and long-term challenges. *In* Uses of By-Products and Wastes in Agriculture. Eds. J Rechcigl and H C MacKinnon. pp 72–90. Am. Chem. Soc., Washington, DC.

Sims J T and Sharpley A N 1998 Managing agricultural phosphorus for water quality protection: future challenges. *In* Soil Testing for Phosphorus: Environmental Uses and Implications. Ed. J T Sims. pp. 41–43. Southern Cooperative Series Bulletin No. 389, SERA-IEG 17, University of Delaware, Newark, DE.

Sims J T, Simard R R and Joern B C 1998 Phosphorus losses on agricultural drainage: historical perspectives and current research. J. Environ. Qual. 27, 277–293.

Smith S J, Sharpley A N, Naney J W, Berg W A and Jones O R 1991 Water quality impacts associated with wheat culture in the Southern Plains. J. Environ. Qual. 20, 244–249.

Uhlen G 1988 Surface runoff losses of phosphorus and other nutrient elements from fertilized grassland. Norwegian J. Agric. Sci. 3, 47–55.

US Department of Agriculture and US Environmental Protection Agency 1999 Unified national strategy for Animal Feeding Operations. March 9, 1999. (http://www.epa.gov/owm/finafost.htm).

US Environmental Protection Agency 1996 Environmental indicators of water quality in the United States. EPA 841-R-96-002. US EPA, Office of Water (4503F), U.S. Govt. Printing Office, Washington, DC.

US Environmental Protection Agency 1998 Water quality criteria and standards plan: priorities for the future. EPA-823-F-98-011. US EPA, Office of Water (4304F), US Govt. Printing Office, Washington, DC.

US Geological Survey 1999 The quality of our nation's waters: Nutrients and pesticides. US Geological Survey Circular 1225, 82 p. USGS Information Services, Denver, CO. http://www.usgs.gov.

Walker J, Alexander D, Irons C, Jones B, Penridge H and Rapport D 1996 Catchment health indicators: an overview. *In* Indicators of Catchment Health: a Technical Perspective. Eds. J Walker and D J Reuters. pp 3–18. CSIRO, Collingwood, Victoria, Australia.

Westerman P W, Donnely T L and Overcash M R 1983 Erosion of soil and poultry manure — a laboratory study. Trans. Am. Soc. Agric. Eng. 26, 1070–1078, 1084.

White R E and Thomas G W 1981 Hydrolysis of aluminum on weakly acidic organic exchangers: implications for phosphorus adsorption. Fertil. Res. 2, 159–167.

Wilkinson R L and Cary J W 1993 Monitoring Soilcare in North East Victoria. Report of the School of Agriculture and Forestry, University of Melbourne, Melbourne, Australia.

Withers P J A, Davidson I A and Foy R H 2000. Prospects for controlling diffuse phosphorus loss to water. Environ Qual. 29, 167–175.

Young T C and DePinto J V 1982 Algal–availability of particulate phosphorus from diffuse and point sources in the lower Great Lakes basin. Hydrobiologia 91: 111–119.

Young R A and Mutchler C K 1976 Pollution potential of manure spread on frozen ground. J. Environ. Qual. 5, 174–179.